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Prescribed burning for control of yellow starthistle (*Centaurea solstitialis*) and enhanced native plant diversity

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Two separate open grassland areas within Sugarloaf Ridge State Park, Sonoma County, CA, were burned for three consecutive years (1993–1995 [Site A] and 1995–1997 [Site B]) for control of yellow starthistle. Burns were conducted in late June to early July following seed dispersal and senescence of desirable grasses and forbs but prior to viable seed production in yellow starthistle. After the first year burn, there was no significant reduction in yellow starthistle cover the following spring and summer. Despite the lack of control, the first year burn reduced the yellow starthistle soil seedbank by 74% and the number of seedlings the following spring by 83%. However, total plant diversity and species richness increased dramatically in the burned areas. This was due primarily to an increase in the number of native broadleaf species. A second burn the next summer (1995–1997 site) reduced seedbank, seedling density, and summer vegetative cover the following year by 94, 92, and 85%, respectively, while maintaining significantly higher native plant cover and richness. A third consecutive summer burn decreased yellow starthistle seedbank and seedling density by 96, 98, and 85%, respectively, in the 1995–1997 burn site. Three consecutive years of burning in the 1993–1995 site reduced yellow starthistle seedbank and seedling density by over 99% and summer vegetative cover by 91%. These results indicate that prescribed burning can be an effective tool for the management of yellow starthistle and can have a long-term benefit on native broadleaf diversity and richness.

Nomenclature: Yellow starthistle, *Centaurea solstitialis* L. CENSO.

Key words: Plant diversity, rangeland, grassland, fire, CENSO.

Yellow starthistle (*Centaurea solstitialis* L.) is a winter annual, native to dry open habitats in southern Europe. It was first introduced into the United States in the 1850s as a contaminant of ballast soil or alfalfa seed (*Medicago sativa* L.) in Oakland, CA (Gerlach 1997; Maddox 1981). Since 1960, it has spread exponentially to infest rangeland, native grasslands, orchards, vineyards, pastures, roadsides, and other areas throughout the western United States (Maddox and Mayfield 1985). Today, yellow starthistle is the most widely distributed weed in California, occupying between 5 and 8 million ha. Although not as widespread in other western states, yellow starthistle is also a major rangeland weed in Oregon, Washington, and Idaho (Roché and Roché 1991).

It is difficult to ascribe a monetary value to the effect of yellow starthistle on rangeland and other noncrop areas. However, it is well recognized that infestations lower forage yield and quality, interfere with grazing, cause problems in forage and crop harvesting, increase livestock and crop management and production costs, and reduce recreational and economic value of infested land. In addition, yellow starthistle causes a lethal neurological disorder in horses known as "chewing disease" (Kingsbury 1964). Yellow starthistle infestations can reduce wildlife habitat and forage, displace native plants, and decrease native plant and animal diversity (Sheley and Larson 1994).

Several methods are used to control yellow starthistle (DiTomaso et al. 1998, Thomsen et al. 1996). These include tillage, mowing, grazing, reseeding, biological control, and herbicides. Many of these methods are restricted in specific environments. In many cases, herbicide use is either prohibited or discouraged, particularly when the management objective is to enhance native broadleaf plant diversity.

Historically, fire has played an important role in the maintenance of many ecosystems, including grasslands (Hatch et al. 1991). Historical fire frequency within the northern Coastal Range of California was analyzed through fire scars on redwoods (*Sequoia* spp.) (Finney and Martin 1992). From this work, 67% of the fire intervals were between 2 and 10 yr. These results are a conservative estimate, as they probably excluded grass fires not intense enough to leave fire-scars on surrounding redwoods. Thus, the success of plant species native to these areas likely depends upon occasional fires to reduce competition, remove thatch, enrich the soil, or scarify seeds. Plant communities in most grassland ecosystems typically show increased plant diversity after fires and reduced diversity following long periods of fire suppression (Kruger 1983).

Sugarloaf Ridge State Park is in the northern Coast Range of California near Sonoma. It is a typical dry, low-elevation woodland with large expanses of open grassland. Since the state acquired the land in the 1920s, fire suppression has been a common practice. Around 1984, yellow starthistle began to spread rapidly through the park, and within 5 yr nearly all open grassland areas were severely infested. In these infested areas, no existing control options were practical or acceptable. Although prescribed burning has been used somewhat successfully to manage other rangeland species, particularly shrubs such as big sagebrush (*Artemisia tridentata* Nutt.), false broomweed (*Ericameria austrotexana* M. Johnst.), and junipers (*Juniperus* spp.) (Mayeux and Hamilton 1988; Rasmussen 1994; West 1991), no studies have previously examined the effect of burning on yellow starthistle control.

The objectives of this study were to determine the effectiveness of multiple-year prescribed burning on yellow starthistle control and seedbank reduction, as well as evaluate the effect of burning on species composition, particularly changes in the diversity and richness of the native plant community. The long-term goal is to provide land managers with an effective and economical means of managing yellow starthistle, while returning the ecosystem to a more healthy condition.

Materials and Methods

Burn Plan

A 14-ha prescribed burn was conducted within a heavily infested grassland site in Sugarloaf Ridge State Park (Sonoma County, CA) in 1993, 1994, and 1995 (Site A). Measurements within Site A were made during the growing season after the second (1994) and third (1995) year of the prescribed burn. A second, 70-ha grassland area at Sugarloaf Ridge State Park was similarly burned in 1995, 1996, and 1997 (Site B). Within Site B, measurements described below were made 1 yr prior to burning and following the first, second, and third year of burning. An adjacent 4-ha unburned control site was separated from the burn area by constructing a firebreak.

The prescribed burns were conducted following senescence of desirable annual grass and broadleaf species but prior to viable seed production in yellow starthistle. This corresponded to the 2 to 5% flowering stage in yellow starthistle (Benefield 1998). Except for the 1997 burn, all prescribed burns were conducted in early July. In 1997, lack of spring moisture resulted in rapid development of yellow starthistle, necessitating a June burn. Following construction of firebreaks, the burns were initiated with a drip torch. A steady backing fire was sustained in most locations. In all locations, the fire carried well (1.2 m flame length), causing nearly complete yellow starthistle mortality. A detailed description of the burn plan was reported by Hastings and DiTomaso (1996a, 1996b).

Fire Temperature at Soil Surface

The temperature at the soil surface during a burn was measured in 1995 by placing Tempil temperature pellets[1] on the ground. Ten tablets, at a range of calibrated melt temperatures between 101 and 399 C, were placed in three randomly selected locations of Site A and B. Burn temperature was determined by observing the highest temperature tablet melted by the fire.

Vegetative Cover and Plant Species Diversity

Three 50- by 50-m pseudoreplicated sampling areas were established within each of the two burn sites and in an adjacent unburned area. All measurements were conducted within these nine sampling areas. Vegetative cover and plant diversity were estimated using a point method: observations were made of plant species intercepting each of 50 points (at 30-cm intervals) along randomly placed 15-m line transects (3 transects per sampling area). Measurements were conducted in spring (late April to early May) prior to yellow starthistle bolting and in summer 1 d prior to the prescribed burn when yellow starthistle was in early flower. Plant diversity was calculated from data obtained in spring using the Shannon–Wiener index (Magurran 1988).

Plant Species Richness

An additional measure of plant diversity was obtained by counting all species present within randomly placed quadrats of 0.0625, 0.25, 1, 4, and 16 m^2 (four subsamples of each size per sampling area). Species richness lines were developed by plotting species numbers by quadrat size on a log-log scale. Both line transect data and quadrat analysis were used to determine the effect of burning on changes in frequency and cover of various species within the study site.

Seed and Seedling Density

Yellow starthistle seedbank was determined in burned and unburned sites after seed dispersal in fall. Seeds from five soil samples per sampling area (5 cm diameter by 5 cm deep) were collected in November and counted following extraction with a water/air elutriator similar to that described by Wiles et al. (1996). Intact achenes were counted using a stereomicroscope.

Densities of yellow starthistle seedlings were evaluated in March, prior to bolting. Seedling numbers were counted in each sampling area in randomly thrown 0.25-m^2 quadrats (10 replicates). Both seed and seedling densities are presented as numbers per square meter.

Soil Characteristics and Light Penetration

Soil moisture and temperature were monitored in both burned and unburned sites. Field measurements of soil moisture (four replicates per sampling area) and temperature at 1, 5, and 10 cm below the soil surface (10 replicates per sampling area) were recorded every month during the 1995 (temperature) and 1996 (moisture and temperature) growing season. Soil temperatures were measured with a probe thermometer[2] and moisture samples were taken 5 cm deep using a pounded corer (5 cm diameter).

Photosynthetically active radiation (PAR, W m^{-2}) was measured with a Decagon Sunfleck Ceptometer[3] 1, 20, and 50 cm above the soil surface at 10 locations in each sampling area of both burned and unburned sites. Measurements were taken in April or May 1995, 1996, and 1997.

Data Analysis

Values for vegetative cover; soil moisture; starthistle seed and seedling densities; and starthistle height, weight, and seedhead production were compared using multiresponse permutation procedures (MRPP), a nonparametric statistical technique that allows comparison of predetermined groups without relying on distributional assumptions (Mielke et al. 1981). When multiple comparisons were required (e.g., comparing vegetative cover of several species classifications among burn treatments), an initial overall comparison was first performed analogous to multiple analysis of variance. If this initial comparison detected significant differences

TABLE 1. Relative vegetative cover and plant diversity in spring (late April to mid-May).[a]

Site	Treatment	Year	Relative cover: YST	Relative cover: Introduced annual grasses	Relative cover: Native perennial grasses	Relative cover: Total grasses	Relative cover: Introduced forbs	Relative cover: Native forbs	Relative cover: Total forbs	Total cover, all species	Shannon–Wiener plant diversity index (− YST)
			%	%	%	%	%	%	%	%	
a. Time series analysis for each site.											
A	2-yr burn	1995	3.1 a	49.5 a	1.5 a	56.9 a	19.3 a	23.6 a	42.9 a	289 a	2.26 a
	3-yr burn	1996	0.7 a	52.9 a	8.3 a	64.4 a	9.9 a	25.7 a	35.6 a	310 a	2.24 a
B	Prior to burn	1995	28.6 a	33.6 a	1.5 bc	44.6 a	41.3 a	6.3 b	47.5 a	261 a	1.68 b
	1-yr burn	1996	12.9 b	39.2 a	0.7 c	46.2 a	30.0 a	23.9 a	53.8 a	302 a	2.23 a
	2-yr burn	1997	2.1 c	40.8 a	3.4 b	50.6 a	19.3 a	28.8 a	48.1 a	215 b	2.08 a
	3-yr burn	1998	1.5 c	31.6 a	10.4 a	45.2 a	20.7 a	33.8 a	54.5 a	278 a	2.30 a
Control	Unburned	1995	30.5 a	33.7 a	3.5 a	46.9 a	37.8 a	12.7 a	50.5 a	253 a	1.70 a
		1996	29.0 a	45.1 a	4.3 a	55.8 a	37.2 a	6.8 a	44.0 a	215 a	1.55 a
		1997	25.8 a	32.3 a	6.8 a	48.8 a	38.1 a	8.4 a	46.5 a	175 a	1.64 a
		1998	29.1 a	30.5 a	10.5 a	51.1 a	41.0 a	7.9 a	48.9 a	258 a	1.73 a
Treatment	Site										
b. Comparison of burn treatments over all years and all sites.											
Unburned	Unburned	all	28.0 a	33.3 b	5.9 a	48.9 a	40.1 a	8.5 b	48.6 a	228 b	1.68 b
	Site B	1995									
1-yr burn	Site B	1996	12.9 b	39.2 ab	0.7 b	46.2 a	30.0 ab	23.8 a	53.8 a	302 a	2.23 a
2-yr burn	Site A	1995	2.9 c	46.9 a	2.0 b	54.4 a	19.7 b	25.2 a	45.0 a	252 ab	2.19 a
	Site B	1997									
3-yr burn	Site A	1996	1.0 c	44.5 ab	9.1 a	56.8 a	14.3 b	29.0 a	43.2 a	297 a	2.26 a
	Site B	1998									

[a] Within each column and each site (1a) or within each column (1b), values followed by the same letter are not significantly different ($\alpha = 0.05$). Shannon–Wiener indices were calculated for individual transects, then the starthistle component was subtracted.

among treatments, then further analysis was performed in order to detect treatment differences within individual species groups. Likewise, MRPP were used for comparing data from multiple years within each burn treatment.

The Shannon–Wiener index is a common method of evaluating plant and animal diversity within a specific area (Buongiorno et al. 1994; Chandler and Peck 1992; Johnson et al. 1996; Stuart et al. 1993). In this study, line transect data were used to calculate Shannon–Wiener plant diversity indices. The contribution of yellow starthistle was subtracted from the total to better represent the diversity of more desirable species.

Soil temperature measurements were compared in series groups (i.e., readings from all three depths at each site were used for multivariate comparison with other sites) using MRPP. PAR measurements were treated similarly. The MRPP subprogram was accessed in PC-ORD version 2.0 (McCune and Mefford 1995).

Results and Discussion

Burn Conditions

The 1995 and 1996 burns generated similar temperatures at the soil surface. Temperatures averaged 231 C in areas previously burned (Site A) and 219 C in the first-year burn area (Site B). Seeds of several plant species were collected from the soil surface and germinated under greenhouse conditions, indicating that the range of temperatures (149 to 302 C) in these prescribed grassland fires was not sufficient to destroy seed viability. In all years, the dried vegetation carried the fire well, with flame lengths of 0.6 to 1.5 m. Burns were conducted at ambient temperatures between 21 and 27 C, relative humidity between 35 and 52%, and light winds ranging between 3 and 8 km h^{-1}.

Yellow Starthistle Cover

Relative cover of yellow starthistle in spring (late rosette stage) was not significantly different among the unburned control site from 1995 through 1998 and Site B in 1995, the year prior to the initial burn (Table 1a). After a single year of burning, starthistle cover decreased significantly (> 40%) in Site B, compared to the previous year and the unburned control in the same year. Despite this reduction of spring cover, the percent cover of yellow starthistle in summer after a 1-yr burn did not differ dramatically from cover in the unburned control or in the same site the year previous to burning (Table 2a). Although the density of yellow starthistle was reduced in burned sites, individual plants were larger by 99 and 320% (dry weight/plant) and produced 73 and 132% more seedheads per plant in the 1-yr and 3-yr burn sites, respectively, compared to unburned control sites (Table 2a).

After two consecutive years of burning, both sites A and B showed reduced yellow starthistle cover compared to the unburned controls and to Site B prior to burning. Compared to the unburned control site, spring cover decreased in the burn sites by approximately 90% in Site A and 92% in Site B (Table 1a), and summer cover decreased by 67% in Site A and 85% in Site B (Table 2a). A third year of

TABLE 2. Vegetative cover and individual plant measurements in summer (late June to early July).[a]

Site	Treatment	Year	Living cover: YST	Living cover: Native perennial grasses	Living cover: Native forbs	Mean YST dry weight/plant	Mean YST height	Mean seedheads/ YST plant
			%	%	%	g	cm	
a. Time series analysis for each site.								
A	2-yr burn	1995	26.2 a	13.1 a	8.0 a			
	3-yr burn	1996	6.9 a	29.8 a	11.6 a	2.21 a	45.2 a	5.1 a
B	Prior to burn	1995	79.3 a	4.7 c	6.0 a			
	1-yr burn	1996	64.7 a	2.3 c	18.7 a	1.37 a	45.1 a	3.8 a
	2-yr burn	1997	8.0 b	16.0 b	9.0 a			
	3-yr burn	1998	11.3 b	29.7 a	3.7 a			
Control	Unburned	1995	80.2 a	2.2 a	6.6 a			
		1996	73.6 a	3.8 a	5.3 a	0.69 a	41.7 a	2.2 a
		1997	52.9 b	15.1 a	4.7 a			
		1998	76.0 a	21.6 a	6.0 a			
Treatment	Sites							
b. Comparison of burn treatments over all years and all sites.								
Unburned	Unburned	all	71.9 a	9.8 bc	5.7 a	—	—	—
	Site B	1995						
1-yr burn	Site B	1996	64.7 a	2.3 c	18.7 a	—	—	—
2-yr burn	Site A	1995	18.9 b	14.3 b	8.4 a	—	—	—
	Site B	1997						
3-yr burn	Site A	1996	8.7 b	29.7 a	11.3 a	—	—	—
	Site B	1998						

[a] Within each column and each site (2a) or within each column (2b), values followed by the same letter are not significantly different ($\alpha = 0.05$). Weight, height, and seedhead counts are compared across all sites for 1996.

burning reduced yellow starthistle cover further: spring cover decreased by 98% in Site A and 95% in Site B, and summer cover decreased by 91% in Site A and 85% in Site B compared to the unburned control. However, reduced levels of infestations in the unburned control area and Site B in 1997 were attributed to an extremely dry spring (Figure 1).

In order to make a robust comparison of treatments (i.e., no burn, 1-yr burn, 2-yr burn, and 3-yr burn), spring (Table 1b) and summer (Table 2b) data for each treatment were combined over all appropriate years. The combined data indicate a significant reduction in yellow starthistle cover in the spring after a single year of prescribed burning and a further significant reduction after a 2-yr burn. In contrast, a significant reduction in summer cover of yellow starthistle did not occur until a second year of burning. A third year of burning further reduced spring and summer yellow starthistle cover, but the differences between the 2-yr and 3-yr burns were not significant.

The reduction in vegetative cover following three consecutive years of burning corresponded to a depletion in the soil seedbank of yellow starthistle (Table 3). After the third burn, the seedbanks in sites A and B were reduced by 99.5 and 96.3%, respectively, compared to the unburned control in the same year. There was a similar reduction in seedling density the following spring. Seedling density decreased 99.6 and 97.9% in sites A and B, respectively, compared to the unburned control site in the same year.

Callihan et al. (1993) reported 9% germination in yellow starthistle achenes, after 6 yr of burial. In contrast, Joley et al. (1992) demonstrated 80% depletion in yellow starthistle seedbank after 1 yr when achenes were dispersed on the soil surface, and 96% depletion after 3 yr. Results presented in Table 3 support those of Joley et al. (1992) and suggest that the longevity of viable seeds under normal field conditions in California may be shorter than previously believed. Nevertheless, a small percentage of seeds likely survive for several years and can potentially lead to reinfestation of previously infested sites, even in the absence of off-site seed recruitment.

These results indicate that prescribed burning can be an effective and economical tool for starthistle suppression in noncrop environments. However, timing is critical to the success of this method. At Sugarloaf Ridge State Park, a 'burn window' opens up after desirable grasses and broadleaf species have completed their reproductive cycle but before yellow starthistle produces seed. Burning at this time should lead to depletion of the yellow starthistle seedbank and to an increase in seedbank, establishment, and competitive ability of desirable species.

Effect of Prescribed Burning on Plant Composition, Diversity, and Species Richness

Between 1995 and 1998 there were no significant changes in relative cover of the major vegetative groups in the unburned control site (Table 1a) despite large fluctuations in seasonal precipitation (Figure 1). Consequently, it was possible to combine data for each treatment over all appropriate years (Table 1b). Combined data indicate that prescribed burning did not have a significant effect on the cover of introduced annual grasses or total forbs (herbaceous dicots and nongraminaceous monocots); however, there was a dramatic change in the proportion of native and introduced forbs. In Site B alone, three consecutive years of prescribed

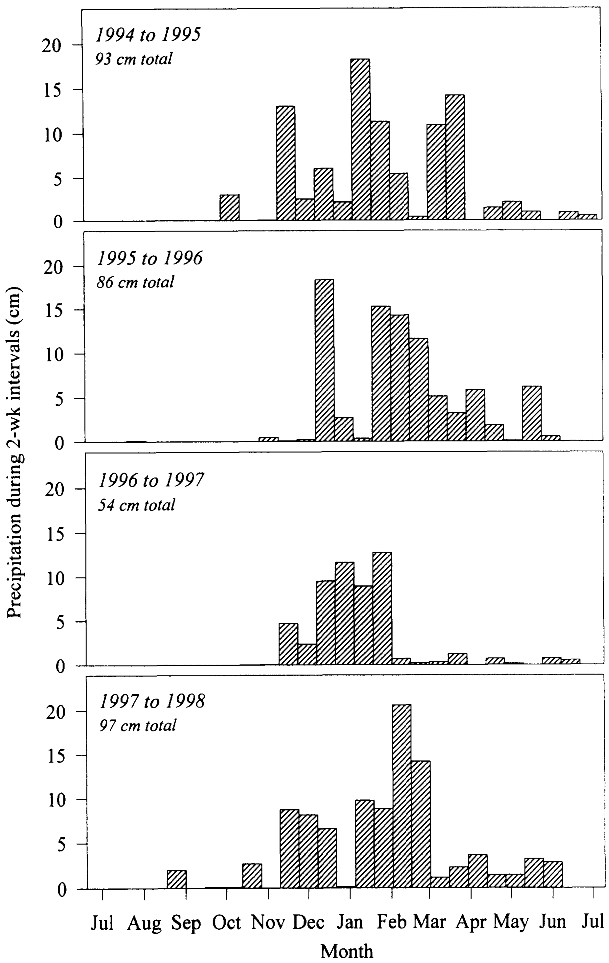

FIGURE 1. Precipitation at Napa, CA, during 2-wk intervals from July 1994 through June 1998.

burning increased native forb cover about five times, boosting the native proportion of total forbs from 13% live canopy to 62% (Table 1a). Over all sites, native forb cover increased from 17% to 67% of total forb cover after 3 yr of burning (Table 1b).

Spring cover of native perennial grasses, primarily purple needlegrass [*Nassella pulchra* (A. Hitchc.) Barkworth], in Site B initially decreased following the first burn, but increased significantly after three consecutive years of burning (Table 1a). December surveys of *Nassella* clumps indicated that cover reduction following the first year burn was likely due to a reduction in the size of bunchgrass clumps rather than to clump mortality (data not shown). Summer cover of native perennial grasses in Site B (Table 2a) showed a similar initial decrease, but increased 3.4 times after 2 yr of burning and by 6.3 times after 3 yr of burning. Although no preburn measurement was taken in Site A (1993), summer cover of native perennial grasses was six times greater than in the unburned control site (1995) after 2 yr of burning (1995), and nearly eight times greater after 3 yr of burning (1996) (Table 2a). Combining data over all sites, summer cover of native perennial grasses was three times higher in 3-yr burn sites than in unburned sites (Table 2b). Interestingly, cover of perennial grasses also showed a large, though nonsignificant, increase from 1995 to 1998 in the unburned control site (Table 2a). Because this increase parallels that in Site B, it is difficult to determine whether burning was the primary

TABLE 3. Within-year comparisons of starthistle seed and seedling counts.[a]

Site	Treatment	Burn year	Seeds in fall after burn	Seedlings in spring after burn
			m^{-2}	
A	3-yr burn	1995	52 c	5 c
B	1-yr burn		2,673 b	230 b
Control	Unburned		10,127 a	1,328 a
B	2-yr burn	1996	421 b	38 b
Control	Unburned		7,042 a	499 a
B	3-yr burn	1997	127 b	19 b
Control	Unburned		3,438 a	910 a

[a] Within each column, values followed by the same letter are not significantly different ($\alpha = 0.05$) for treatments in the same year.

reason for the increase in Site B. Other studies, however, have shown increases in purple needlegrass following grassland fires (Fossum 1990; Hatch et al. 1991).

A comparison of Shannon–Wiener diversity indices (Table 1a and 1b) indicates that unburned sites in all years maintained a lower level of plant diversity. Interestingly, a single burn in Site B did not significantly reduce summer yellow starthistle cover (Table 2a), but dramatically increased plant diversity compared to the unburned site or Site B the year prior to burn.

Data from quadrat analyses were used as a second method of evaluating the effect of prescribed burning and yellow starthistle on plant diversity. Log-log transformations of these data indicated total species richness was consistently and statistically greater in burned sites compared to unburned control sites (data not shown). These data support the results obtained using the Shannon–Wiener index. Separate species richness analysis of grasses and forbs found no differences in grasses among treatments in each year (data not shown). In contrast, a higher number of forb species in all quadrat sizes accounted for increased species richness in the burned sites in each year (Figure 2). This increase in species richness of forbs was independent of the number of consecutive burns.

Using both the vegetative cover data obtained from line transects and frequency data from quadrat analysis, it was possible to develop a list of species demonstrating significant expansion (Table 4) or decline (Table 5) following prescribed burning. The predominant species showing increased prevalence in the burned sites were wild oat (*Avena fatua* L.) and native forbs, particularly members of the Fabaceae. Wild oat averaged over four times higher relative cover in burned vs. unburned sites, as did the native forb *Linanthus bicolor* (Nutt.) E. Greene. Major native legume species (Fabaceae) consistently increased in abundance in burned sites, including *Lotus wrangelianus* Fischer et C. Meyer (7.7 times greater relative cover), *Lupinus nanus* Benth. (23 times greater), and *Trifolium gracilentum* Torrey et A. Gray (2.4 times greater). Fifty-two percent of the species demonstrating a dramatic increase in vegetative cover and frequency were California natives. By comparison, most species declining in abundance and frequency following prescribed burning were introduced grasses and forbs. In addition to dramatically reducing the abundance of yellow starthistle, burning also significantly reduced the relative

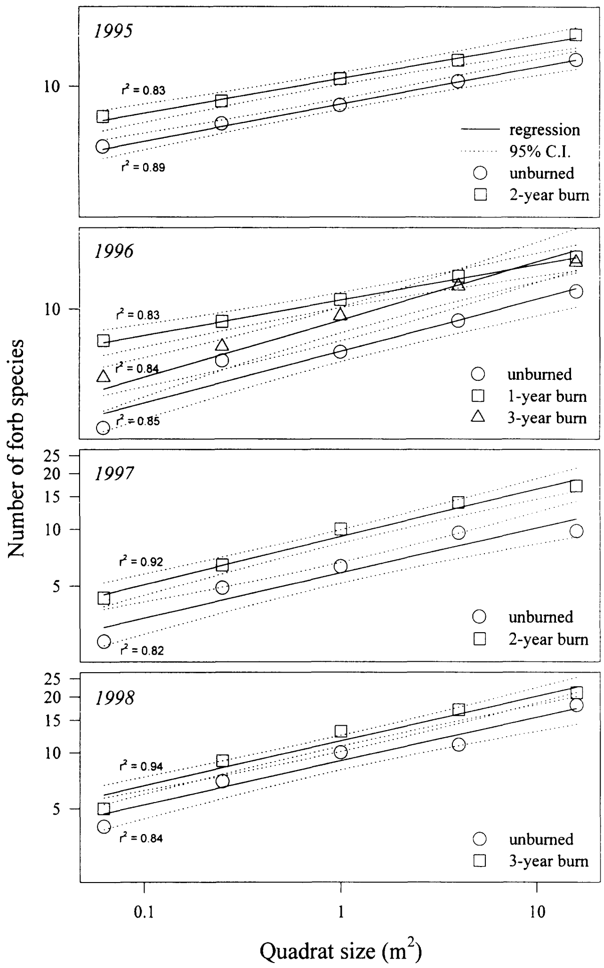

FIGURE 2. Species richness of forbs plotted against quadrat size. Symbols represent the mean of quadrat sizes for each sampling area. Solid lines represent the regression of means for each treatment. Dashed lines indicate the 95% confidence interval (CI) for each regression.

cover of false brome [*Brachypodium distachyon* (L.) P. Beauv.] by an average of 75%, ripgut brome (*Bromus diandrus* Roth) by 97%, and soft chess (*Bromus hordeaceus* L.) by 60%. Overall, only 8% of the species declining after prescribed burning were natives. Of these, purple needlegrass decreased only after the first year of burning and increased after subsequent burns. *Clarkia purpurea* (Curtis) Nelson et J.F. Macbr. is a late-season native herbaceous flowering plant that showed a decrease in abundance on burned sites. Summer burns probably reduced seed production in this species.

Possible Factors Influencing Enhanced Native Plant Diversity

From the results presented in Tables 1 and 2, we speculate that the increase in native plant diversity is not directly correlated with the decrease in yellow starthistle cover. After a single burn, the spring cover, but not the summer cover, of yellow starthistle declined significantly. Comparatively, a single year of prescribed burning dramatically increased native forbs and this level of increase was maintained after a second and third year of burning. We postulate that burning may enhance native plant diversity by removing the thatch layer. At Sugarloaf Ridge State Park, the soil surface in unburned sites is covered with a dense, persistent thatch 1 to 4 cm thick, whereas the ground in burned sites was relatively bare under the plant canopy. It is possible that thatch accumulation, resulting from years of fire suppression, is a primary restraint on native forbs. Knapp and Seastedt (1986) note that species richness decreases in grasslands primarily because of the development of a thick litter layer, and Vogl (1974) suggests that burning off litter permits denser growth by existing species.

It seems likely that presence or absence of thatch may affect several abiotic factors, including PAR, soil temperature, and soil moisture. In previously burned plots, PAR at ground level (1 cm) averaged 39% of ambient (above-canopy) PAR, whereas PAR at ground level in unburned plots averaged only 11% of ambient PAR (Table 6). There were no significant differences between PAR 50 cm above the soil surface in the burned and unburned sites. In 1996, PAR was significantly reduced 20 cm above the soil surface in the unburned plot, but in all other years there were no significant differences.

Likewise, soil temperature 1, 5, and 10 cm deep in burned plots was consistently higher than in unburned plots from March through April in 1995 and May through July in 1996 (Figure 3). Interestingly, this increased soil temperature in the burn site did not correspond with a reduction in surface soil moisture. At five separate 3-wk evaluation intervals between March and June 1996, no significant differences in soil water content were measured among 1-yr, 3-yr, and unburned control sites (data not shown).

Although these measurements do not consider all possible explanations for the increased abundance of native forbs, they suggest two possible means by which burning may improve the local environment for establishment of desirable native species relative to yellow starthistle. First, seeds of many endemic native forbs may have a high light requirement for germination and for survival of young seedlings. Thatch removal could enhance the germination and establishment of these endemic species. Second, native species may also require higher soil temperatures for germination. Thatch removal would permit earlier soil warming in spring, thus encouraging greater native forb germination and establishment in these grassland ecosystems. Furthermore, the earlier growth could increase grassland productivity by extending the growing season (Vogl 1974). Similar explanations for increased plant diversity following burning have been suggested by other researchers (Hassan and West 1986; Heady and Child 1994; Vogl 1974). It is also likely that removal of the heavy thatch layer with a prescribed burn would release nutrients for rapid growth and increased survivorship of many grassland species (Evans 1960; Tilman 1987; Vogl 1974). These nutrients otherwise would be released only slowly by thatch decay. Alternatively, a thick covering of thatch may provide an environment conducive to the survival of pathogens that may increase the mortality of native forbs, particularly in the seedling stage.

These results indicate that periodic burning in California grassland ecosystems can reduce yellow starthistle infestations, enhance native plant diversity, and increase survival of competitive native perennial grasses. Prescribed burning is not without risk: grassland fires can negatively affect air quality, compromise establishment of biocontrol agents, and lead to catastrophic wildfires should they escape containment. Nevertheless, fire has long been an important factor

Table 4. List of species increasing in vegetative cover and frequency in burn sites compared to unburned sites.[a]

	Site A					Site B				
Burn treatment	Year	Species	Vegetation type[b]	N/I[c]	Family	Year	Species	Vegetation type	N/I	Family
1-yr burn						1996	*Avena fatua*	AG	I	Poaceae
							Briza minor	AG	I	Poaceae
							Gastridium ventricosum	AG	I	Poaceae
							Plagiobothrys nothofulvus	F	N	Boraginaceae
							Petrorhagia nanteuilii	F	I	Caryophyllaceae
							Silene gallica	F	I	Caryophyllaceae
							Lathyrus cicera	F	I	Fabaceae
							Lotus wrangelianus	F	N	Fabaceae
							Lupinus nanus	F	N	Fabaceae
							Trifolium gracilentum	F	N	Fabaceae
							Trifolium microdon	F	N	Fabaceae
2-yr burn	1995	*Avena fatua*	AG	I	Poaceae	1997	*Avena fatua*	AG	I	Poaceae
		Vulpia myuros	AG	I	Poaceae		*Brachypodium distachyon*	AG	I	Poaceae
		Astragalus gambelianus	F	N	Fabaceae		*Petrorhagia nanteuilii*	F	I	Caryophyllaceae
		Lotus wrangelianus	F	N	Fabaceae		*Silene gallica*	F	I	Caryophyllaceae
		Lupinus nanus	F	N	Fabaceae		*Lotus wrangelianus*	F	N	Fabaceae
		Trifolium gracilentum	F	N	Fabaceae		*Lupinus nanus*	F	N	Fabaceae
		Trifolium microdon	F	N	Fabaceae		*Linanthus bicolor*	F	N	Polemoniaceae
		Erodium circutarium	F	I	Geraniaceae		*Galium divaricatum*	F	I	Rubiaceae
		Linanthus bicolor	F	N	Polemoniaceae					
3-yr burn	1996	*Aira caryophyllea*	AG	I	Poaceae	1998	*Avena fatua*	AG	I	Poaceae
		Avena fatua	AG	I	Poaceae		*Hemizonia congesta*	F	N	Asteraceae
		Briza minor	AG	I	Poaceae		*Polycarpon tetraphyllum*	F	I	Caryophyllaceae
		Nassella pulchra	PG	N	Poaceae		*Lotus wrangelianus*	F	N	Fabaceae
		Polycarpon tetraphyllum	F	I	Caryophyllaceae		*Lupinus nanus*	F	N	Fabaceae
		Lotus wrangelianus	F	N	Fabaceae		*Trifolium gracilentum*	F	N	Fabaceae
		Trifolium gracilentum	F	N	Fabaceae		*Erodium cicutarium*	F	I	Gernaniaceae
		Trifolium microdon	F	N	Fabaceae		*Dichelostemma* sp.	F	N	Liliaceae
		Erodium cicutarium	F	I	Geraniaceae		*Linanthus bicolor*	F	N	Polemoniaceae
		Linanthus bicolor	F	N	Polemoniaceae					
		Galium divaricatum	F	I	Rubiaceae					

[a] Species listed are those that make up at least 1.5% of relative vegetative cover, that are found in at least 20% of 0.0625-m^2 quadrats in the sites under comparison, and that showed an increase of at least 75% in both cover and frequency in burned sites.
[b] AG, annual grass; PG, perennial grass; F, forb.
[c] N, native; I, introduced.

TABLE 5. List of species decreasing in vegetative cover and frequency in burn sites compared to unburned sites.[a]

	Site A					Site B				
Burn treatment	Year	Species	Vegetation type[b]	N/I[c]	Family	Year	Species	Vegetation type	N/I	Family
1-yr burn						1996	*Brachypodium distachyon*	AG	I	Poaceae
							Bromus diandrus	AG	I	Poaceae
							Bromus hordeaceus	AG	I	Poaceae
							Nassella pulchra	PG	N	Poaceae
							Vulpia myuros	AG	I	Poaceae
2-yr burn	1995	*Bromus diandrus*	AG	I	Poaceae	1997	*Bromus diandrus*	AG	I	Poaceae
		Bromus hordeaceus	AG	I	Poaceae		*Bromus hordeaceus*	AG	I	Poaceae
		Lolium perenne	PG	I	Poaceae		*Vulpia myuros*	AG	I	Poaceae
		Centaurea solstitialis	F	I	Asteraceae		*Centaurea solstitialis*	F	I	Asteraceae
							Clarkia purpurea	F	N	Onagraceae
3-yr burn	1996	*Brachypodium distachyon*	AG	I	Poaceae	1998	*Brachypodium distachyon*	AG	I	Poaceae
		Bromus diandrus	AG	I	Poaceae		*Bromus diandrus*	AG	I	Poaceae
		Bromus hordeaceus	AG	I	Poaceae		*Bromus hordeaceus*	AG	I	Poaceae
		Centaurea solstitialis	F	I	Asteraceae		*Centaurea solstitialis*	F	I	Asteraceae
		Vicia sativa	F	I	Fabaceae					
		Geranium dissectum	F	I	Geraniaceae					

[a] Species listed are those that make up at least 1.5% of relative vegetative cover, that are found in at least 20% of 0.0625-m^2 quadrats in the sites under comparison, and that are at least 75% more common in both cover and frequency in unburned sites.
[b] AG, annual grass; PG, perennial grass; F, forb.
[c] N, native; I, introduced.

Table 6. Photosynthetically active radiation (PAR) at three heights above the soil surface during spring.[a]

Year	Date	Site	Treatment	Ambient PAR 50 cm	20 cm	1 cm
				%		
1995	26 May	A	2-yr burn	90 a	63 a	32 a
		Control	Unburned	96 a	70 a	10 b
1996	29 April	A	3-yr burn	100 a	87 a	37 a
		B	1-yr burn	99 a	82 a	47 a
		Control	Unburned	100 a	58 b	8 b
1997	13 May	B	2-yr burn	92 a	74 a	39 a
		Control	Unburned	100 a	83 a	16 a

[a] Within each column and each year, values followed by the same letter are not significantly different ($\alpha = 0.05$).

in the development and continuance of most grassland systems (Vogl 1974). Provided burns are conducted at the appropriate time, they can encourage native species, and they show great promise for control of some late-season noxious weeds such as yellow starthistle. It is important to note, however, that the effect of burning is temporary. The Sugarloaf Ridge grassland ecosystem, in its present state of dominance by introduced annual grasses, is susceptible to reinfestation by yellow starthistle. With continuing suppression of natural burns, it is likely that this ecosystem will eventually revert to preburn conditions.

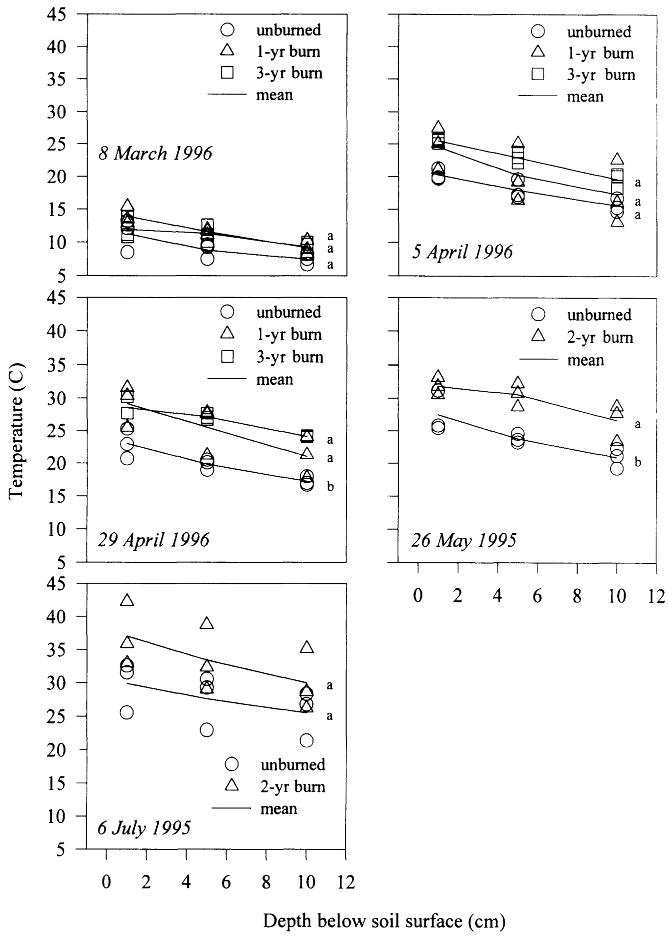

Figure 3. Soil temperature at three depths in unburned, 1-yr, 2-yr, and 3-yr burn sites in 1995 and 1996. Symbols represent all sampling area data points. Lines intercept the mean for each group of data points for each treatment. Line groups followed by the same letter are not statistically different ($P < 0.05$).

Sources of Materials

[1] Tempil temperature pellets, Tempil Div., Big Three Industries, So. Plainfield, NJ 07080.

[2] Air conditioner thermometer, NAPA Temp Products, Cockeysville, MD 21030.

[3] Sunfleck Ceptometer, Decagon Devices, Inc., Pullman, WA 99163.

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